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Nitrogen losses to the environment following food-based digestate and compost applications to agricultural land

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Abstract

The anaerobic digestion of food waste for energy recovery produces a nutrient-rich digestate which is a valuable source of crop available nitrogen (N). As with any 'new' material being recycled to agricultural land it is important to develop best management practices that maximise crop available N supply, whilst minimising emissions to the environment. In this study, ammonia (NH₃) and nitrous oxide (N₂O) emissions to air and nitrate (NO₃⁻) leaching losses to water following digestate, compost and livestock manure applications to agricultural land were measured at 3 sites in England and Wales. Ammonia emissions were greater from applications of food-based digestate (c.40% of total N applied) than from livestock slurry (c.30% of total N applied) due to its higher ammonium-N content (mean 5.6 kg/t compared with 1-2 kg/t for slurry) and elevated pH (mean 8.3 compared with 7.7 for slurry). Whilst bandspreading was effective at reducing NH₃ emissions from slurry compared with surface broadcasting it was not found to be an effective mitigation option for food-based digestate in this study. The majority of the NH₃ losses occurred within 6 hours of spreading highlighting the importance of rapid soil incorporation as a method for reducing NH₃ emissions. Nitrous oxide losses from food-based digestates were low, with emission factors all less than the IPCC default value of 1% (mean 0.45 ± 0.15%). Overwinter NO₃⁻ leaching losses from food-based digestate were similar to those from pig slurry, but much greater than from pig farmyard manure or compost. Both gaseous N losses and NO₃⁻ leaching from green and green/food composts were low, indicating that in these terms compost can be considered as an 'environmentally benign' material. These findings have been

used in the development of best practice guidelines which provide a framework for the responsible use of digestates and composts in agriculture.

Capsule

Field measurements of nitrogen losses as ammonia, nitrous oxide and nitrate leaching have been used to develop best practice guidance for food-based digestate and compost use in agriculture.

Keywords: Emissions; nitrate leaching; ammonia; nitrous oxide; digestate

1. Introduction.

The United Kingdom generates around 14 million tonnes of food waste each year, the highest rate in the European Union, which in total generates nearly 90 million tonnes; the quantity produced by the different member states depends on numerous factors such as cultural practices, climate, diet and socio-economic conditions (EC, 2010). A large proportion of this waste is disposed of to landfill, with the UK sending around 8 million tonnes of biodegradable municipal waste (including food waste) to landfill every year (Defra, 2016). Redirecting this material away from landfill will significantly reduce greenhouse gas (GHG) emissions, in particular methane (CH₄), which has a global warming potential around 25-fold greater than carbon dioxide (CO₂), thereby contributing to GHG reduction targets. To this end, the EU Landfill Directive states that by 2020 the amount of biodegradable municipal waste disposed of in landfill sites must be reduced by 65%, compared with 1995 levels (EC, 1999).

As part of the UK's commitment to reduce GHG emissions and to meet EU renewable energy targets, policies and strategies have been implemented (DECC/Defra, 2011) to increase the treatment of food waste through anaerobic digestion (AD; Styles *et al.*, 2016) which, as well as providing renewable energy, generates a nutrient-rich digestate or 'biofertiliser'. The most recent estimates suggest that by 2020 around 5 Mt of the 7 Mt of food waste currently sent to landfill could be available for digestion annually in the UK (DECC/Defra, 2011). However, AD is not without its problems. In a recent study of the UK biogas sector, Styles *et al.* (2016) found that whilst biogas energy has a lower GHG intensity than fossil fuels, it can increase acidification and eutrophication burdens. Life-cycle assessment (LCA) studies have highlighted the fact that the environmental outcomes are very sensitive to factors such as feedstock type, fugitive emissions, biomethane use, energy conversion efficiency and digestate management. Nevertheless, Styles *et al.* (2016)

concluded that to maximise the potential for GHG abatement, climate change policies should encourage digestion of food waste whilst restricting digestion of crop inputs and wastes that could be used as animal feed.

The AD sector in the UK has been developing rapidly, with a 34% increase in the total number of operational AD plants between 2012 and 2013, and an increase of 51% in the tonnage of organic material processed (WRAP, 2014). Over 2 million tonnes of digestate were applied to agricultural land in the UK in 2013, supplying a total of 92 kt nitrogen (N). Although this is only half the total N supplied by composts, c.80% of the total N within food-based digestate is readily available (WRAP, 2016) i.e. in the form of ammonium-N ($\text{NH}_4\text{-N}$). A regular survey of the organics processing industry has been undertaken since the mid-1990s; in 2012 the survey indicated that there had been little year-on-year change in the UK composting sector, with a total of 3.5 million tonnes (fresh weight) produced in 2012, 68% of which was recycled to agricultural land, supplying 192 kt N.

When applying organic materials such as digestate and compost to land, it is essential that their application, agricultural or otherwise, is not harmful to the environment (i.e. to soil, water and air quality) or human health. The European Nitrogen Assessment (Sutton *et al.*, 2011) highlighted how the overall environmental costs of all N losses in Europe (estimated at €70–€320 billion per year at current rates) outweigh the direct economic benefits of N in agriculture, due largely to loss of air and water quality. The land application of organic materials therefore needs to be carefully managed to maximise their crop available nutrient value and minimise their impact on the wider environment. Food-based digestate is of particular interest being a new, less well understood material which will vary in its compositional characteristics depending on the type of food waste used to produce it (e.g. vegetable-based feed stocks produce digestate with lower N concentrations than those produced from mixed food wastes). It generally has a relatively high total N content of around 5 kg/m^3 (WRAP, 2016) compared with c. 3 kg/m^3 for cattle slurry and c. 4 kg/m^3 for pig slurry (Defra, 2010) and might therefore be expected to have the potential for greater N losses to the environment than livestock manures or manure-based digestates.

Ammonia (NH_3) emissions to air contribute to acid deposition and can cause eutrophication of sensitive ecosystems; in addition, NH_3 reacts with acids in the atmosphere to form particulate matter which may pose a threat to human health (Webb *et al.*, 2004). The amount and rate of NH_3 release following land spreading depends on a range of organic material (e.g. pH, readily available N, dry matter), spreading (e.g. application rate, method and timing), soil (e.g. moisture content) and environmental (e.g. temperature, wind speed, rainfall) factors (Nicholson *et al.*, 2013). There is an extensive body of research in the UK (and elsewhere) on NH_3 emissions following land application of

livestock manures and slurries (see for example Pain *et al.*, 1989; Chambers *et al.*, 1997; Sommer *et al.*, 1997; Huijsmans *et al.*, 2001; Misselbrook *et al.*, 2002; Webb *et al.*, 2004), which has been used to populate the National Ammonia Emissions Inventory (Misselbrook *et al.*, 2015) and provide guidance for farmers to minimise NH₃ emissions from manures in the UK (Defra, 2009) and elsewhere. Research has also been undertaken using manure-based and crop-based digestates applied to land (e.g. Rubaek *et al.*, 1996; Wulf *et al.*, 2002); however, very little information is available for food-based digestates produced and applied under conditions pertinent to the UK. A recent study by Tiwary *et al.* (2015) showed that surface applied food-based digestate applications led to NH₃ losses of 35-65% of the total N applied in the week following application, with an abatement of 85% achieved if the material was incorporated into the soil immediately following application. However, this study was undertaken at a field site in India and used digestates that may not be comparable with those currently produced in the UK. Composts produced from green wastes such as grass clippings and hedge trimmings (green compost) or from a mixture of green and food wastes (green/food compost) tend to be applied to agricultural soils for soil conditioning purposes (WRAP, 2016), although they also contain valuable amounts of plant available nutrients. The low readily available N content of composts (generally <5% of total N; Defra, 2010) would suggest that NH₃ losses following land spreading are also likely to be low, although there is little evidence currently available to support this assertion.

Nitrous oxide (N₂O) is a greenhouse gas with a global warming potential c.300-fold greater than carbon dioxide (IPCC, 2006). The UK Greenhouse Gas Emissions Inventory (2014) estimated that c.70% of N₂O produced in the UK comes from agriculture (Brown *et al.*, 2016), of which the majority (75%) is emitted from soils following N applications/returns (e.g. manufactured fertiliser N, crop residue incorporation organic materials and urine from grazing returns) to land. Around 17% of agricultural N₂O is emitted indirectly from soils following re-deposition of emitted NH₃ and from leached nitrate (NO₃⁻) (Brown *et al.*, 2016). As with NH₃, there is little information available on N₂O losses following food-based digestate and compost applications to agricultural land. The current IPCC Tier 1 default emission factor (EF) for N₂O losses from animal manure, compost, sewage sludge and other organic N additions (e.g. digestates) is 1% of the total N applied (IPCC, 2006). By way of comparison, Tiwary *et al.* (2015) found that N₂O emissions from food-based digestate were 4-10% of the total N applied, which is much higher than the default IPCC EF of 1%, although these measurements were made in India under very different soil and climatic conditions from those in the UK. Measurements in Scotland showed that cumulative N₂O emissions following green compost applied at 35, 100 and 200 t/ha ranged from 0.32 to 4.54 kg N₂O-N/ha/yr, with the higher values measured following the 200 t/ha application in the wet spring of 2008 (Ball *et al.*, 2014). Although

compost application rates were very high in this experiment (35-200 t/ha compared with a more typical rate of 30 t/ha), the maximum N₂O EFs were still only around 1% of the total N applied.

There is still much uncertainty over the factors which control N₂O emissions from food-based digestates following application to agricultural land. For example, Pezzolla *et al.* (2012) found that applying food –based digestate to a UK grassland did not increase emissions compared to the untreated control, although measurements were made during an exceptionally dry growing season. In contrast, an incubation study showed much larger emissions from food-based digestate compared to ammonium sulphate applied under high soil moisture conditions (Koester *et al.*, 2011). A later study under similar conditions found emissions were twice as high from cattle slurry than from food-based digestate (Koester *et al.*, 2015). Following a laboratory incubation study using food-based and other digestates, Rigby & Smith (2013) concluded that “the significance and influence of the interaction between soil type and digestate stability and physical properties on denitrification processes in digestate-amended soils require urgent investigation.”

Additionally, poorly-managed applications of digestate and compost to land have the potential to impact on water quality as a result of NO₃⁻ and other pollutant losses in drainage water. Indeed the 1991 EU Nitrates Directive (91/676/EEC) aims to protect water quality across Europe by preventing NO₃⁻ from agricultural sources from polluting ground and surface waters by promoting good farming practices. Again, there is a substantial body of evidence on NO₃⁻ losses following manure application to agricultural land (e.g. Chambers *et al.*, 2000; Smith *et al.*, 2002), but very few field studies available for food-based digestates. Goberna *et al.* (2011) reported that digestate application to soil doubled NO₃⁻ leaching compared with animal manure applied at an equivalent N rate (80 kg N/ha), however this was a pot-based study using manure-based digestate. Another pot experiment using manure-based digestate applied to grass showed that there was less potential for NO₃⁻ leaching losses compared with undigested slurry or mineral fertiliser (Walsh *et al.*, 2012).

The study described in this paper was part of a wider programme of work to establish a robust scientific evidence base on the nutrient supply properties of digestate and compost applications to land in the UK (Bhogan *et al.*, 2016). The objective of the work described in this paper was to measure N losses following autumn and spring applications of food-based digestate, green/food compost and green compost to agricultural land via NH₃ volatilisation and N₂O emissions to air, and NO₃⁻ leaching to water, and to compare these losses with those from livestock manures. The effect of application techniques (i.e. surface broadcast compared with band spreading) were also assessed to establish the effectiveness of this potential mitigation technique. The findings will be used to develop best management practices that maximise crop nutrient utilisation, whilst minimising environmental emissions of NH₃, N₂O and NO₃⁻.

2. Materials and methods

2.1 Experimental sites

Experimental sites were established at 3 locations on a range of contrasting soil types and agroclimatic zones (Table 1). To characterise each site, representative topsoil samples (0-15cm at the arable site – Wensum - and 0-7.5cm at the grassland sites) were taken prior to the start of the experiment and analysed for pH, sand (%), clay (%), silt (%), total N and organic carbon (C) using standard methodologies (Anon., 1986), with the results shown in Table 1.

2.2 Treatments and design

The organic materials used were food-based digestate, compost, green or green/food compost, green compost, solid farmyard manure (FYM) and livestock slurry. All organic materials were surface broadcast, with additional treatments where food-based digestate and slurry were applied using a bandspreading technique.

At WE, the only site where leaching losses to water were quantified, treatments were applied in August 2011 (autumn), and then repeated in February 2012 (spring) on different experimental plots; at the other sites, only spring applications (March 2012 at NW and May 2012 at PW) were evaluated. For practical reasons, the spring and autumn treatments were grouped separately, with each group of plots having an untreated control. Each treatment was replicated three times and arranged in a randomised block design. Plot sizes were 3-7 m wide by 8-15m long.

Cattle FYM and slurries were used at NW and PW, and pig FYM and slurry at WE. The green compost (PW and NW) or green/food compost (WE) and food-based digestates were sourced from commercial enterprises local to the experimental sites, with the digestates from AD plants using mainly commercial and municipal food wastes as a feedstock. The liquid materials (livestock slurry and food-based digestate) were applied using a specially designed small plot applicator and the solid materials (FYM and compost) were applied by hand (at the NW site the liquid organic materials were also applied by hand). To be representative of commercial practice, at the grassland sites the bandspread liquid material applications were by trailing shoe with 20cm spacing between the bands and at WE, the bandspread applications were by trailing hose with 30cm spacing between the bands. At WE, the autumn applied broadcast pig slurry and food-based digestate were incorporated into the soil within 6 hours of application, using a rotavator to comply with Nitrate Vulnerable Zone (NVZ) rules (SI, 2008; WSI, 2008).

The amount of organic material applied to each plot was weighed and recorded, so that the application rate to each plot could be accurately determined, with the aim of applying the materials

at typical agronomic rates i.e. 20-30 m³/ha digestate, 30-40 m³/ha livestock slurry, 20 t/ha compost and 25-30 t/ha FYM. Organic material application rates and N loadings are shown in Table 2.

2.3 Organic material analysis

At each site and application timing, a representative sample of each organic material type from each experimental block was taken (c.2 litres or 2 kg per block), giving three replicate samples of each material per site. These samples were analysed for pH, dry matter (DM), total N and NH₄-N using standard methodologies (Anon., 1986).

2.4 Crop management

The grass at PW and NW was permanent pasture; the wheat crop at WE was grown according to best farm practice using commercially recommended seed rates, with crop protection products applied as needed and according to good agricultural practice to control weeds, pests and diseases equally across all treatments. No manufactured N fertiliser was applied to the organic material treatments, but all treatments, including the untreated control, had manufactured fertilisers (P, K and S) applied based on the requirements of the untreated control (Defra, 2010).

2.5 Ammonia emissions

Wind tunnels were used to assess NH₃ emissions from the livestock manure treatments at each site, based on the design developed by Lockyer (1984) and as recently used by Tiwary *et al.* (2015) and Misselbrook *et al.* (2014). Each wind tunnel consisted of two parts; a transparent polycarbonate canopy (2.0 m x 0.5 m) which covered the plot area, and a stainless steel duct housing a fan which drew air through the canopy at a speed of 1 m/s; an anemometer measuring the wind speed, which was recorded using a pulse counter. A sub-sample of the air entering and leaving the tunnel was drawn through absorption flasks containing 80 ml of 0.02 M orthophosphoric acid. The absorption flasks were changed after 24 hours and then daily for 7 days. The loss of NH₃ from beneath each tunnel was calculated as the product of air flow through the tunnel and the difference between the concentrations of NH₃ in the air entering (i.e. the background NH₃ concentration) and leaving the tunnel as follows:

$$\text{NH}_3 \text{ loss} = ((b/a)_{\text{outlet}} - (b/a)_{\text{inlet}}) * c \quad (1)$$

where, a is the volume of air sampled by each acid trap, b is the quantity of NH₃-N in each trap over the sampling period and c is the volume of air drawn through each tunnel. The rate of loss was calculated over each time period so that the pattern of loss could be quantified and cumulative losses were then calculated by summing over all sampling periods.

2.6 Nitrous oxide emissions

Nitrous oxide emissions were measured using the static chamber technique (Chadwick *et al.*, 2014), from three replicate plots per treatment, using 5 chambers per plot (giving a total of 15 replicate chambers per treatment). Each chamber had dimensions of 40 cm x 40 cm square and was 25 cm tall, giving a soil surface area coverage of 0.16 m². The chambers were installed immediately after organic material application and positioned in a 5 cm deep slot cut in the soil. The chambers were designed to completely enclose growing arable crops and grassland, without damage, with chamber extensions fitted to enable measurements to be taken from mature cereal and grass crops. On each sampling occasion, the chambers were covered for at least 40 minutes before the headspace was sampled. Sampling was normally conducted between 10 am and 2 pm, as previous studies have shown that emissions at this time of day approximate to average diurnal emission rate (Clayton *et al.*, 1994). The samples were transferred to evacuated vials prior to Gas Chromatography analysis using an Electron Capture Detector. To verify the assumption of linear gas accumulation within a chamber's headspace, 3 chambers were selected on each sampling occasion from which a time series of headspace samples was taken every 15 minutes up to 60 minutes after closure. The following steps were taken to help ensure that linearity in gas accumulation was achieved, by (i) ensuring an air-tight seal between the chamber and soil; (ii) ensuring an air-tight seal between the chamber and lid; (iii) using 'large' chambers to provide as much headspace as practically possible, whilst retaining analytical sensitivity.

Data from previous UK studies have indicated that c.75% of total *direct* N₂O emissions are likely to occur in the first 4-6 weeks following slurry application. Therefore the sampling strategy was weighted accordingly, with c.50% of sampling events carried out during the (likely) period of highest N₂O fluxes (i.e. 4-6 weeks after application), giving a total of at least 30 measurements over a 12 month period.

2.7 N leaching losses

At WE, N leaching losses to water were measured from the plots receiving autumn applications of organic materials, using Teflon cup water samplers. Five water samplers were installed on each plot to a depth of 90 cm. Samples of soil water were collected every 2 weeks or after 50 mm drainage, whichever occurred sooner, throughout the drainage period and analysed for NO₃⁻ and NH₄-N using standard methodologies (Anon, 1986). Drainage estimates were obtained using IRRIGUIDE (Bailey & Spackman, 1996) and were combined with the pollutant concentration data to calculate N losses in drainage water.

259

260 2.8 Other measurements

261 Soil samples for mineral nitrogen (SMN) determination were taken periodically (from 0-10 cm depth)
262 throughout the experimental period to quantify changes in soil mineral N following the organic
263 material applications that could influence gaseous N losses. Soil moisture content measurements
264 (per block) were also made on each N₂O sampling occasion, and converted to water filled pore space
265 (WFPS) using the following formula:

$$266 \quad \text{WFPS (\%)} = \text{Soil moisture (\% w/w)} \times \text{BD} / (1 - (\text{BD} / 2.65)) \quad (2)$$

267 Where BD is the bulk density of the studied soil (g/cm³) and 2.65 g/cm³ is the bulk density of mineral
268 soils.

269 Daily rainfall and mean air and soil temperature (at 5cm depth) data were measured at each site or
270 obtained from a nearby meteorological station.

271 2.9 Statistical analysis

272 At each experimental site, conventional analysis of variance (ANOVA) comparisons were undertaken
273 between the different treatments in terms of cumulative N₂O and NH₃ emissions to air, and NO₃⁻ and
274 NH₄-N losses to water, with comparison of P statistics (quoted in the text). A separate ANOVA was
275 carried out at each site, after which *post-hoc* testing was undertaken to evaluate which treatment
276 means were different from each other using a Duncan's multiple range test (using Genstat version
277 12; VSN International Ltd, 2010).

278 Additionally, the pooled data were analysed to assess effects of N₂O and NH₃ emissions to air (*i.e.* a
279 cross-site analysis ANOVA was undertaken). This enabled us to assess whether a particular
280 treatment had had a statistically significant effect at each site, and across *all* study sites. The ANOVA
281 also indicated if there were significant interactions between sites and treatments.

282 3. Results

283 3.1 Organic material analysis

284 The composition of the organic materials applied at each site and application timing is shown in
285 Table 2. As expected the food-based digestate had higher total N contents (mean 7.0 kg/t across all 4
286 site/timings) than the pig slurry (mean 2.8 kg/t at WE) or cattle slurry (mean 2.4 kg/t at NW and PW).
287 The NH₄-N contents of the food-based digestate and pig slurry (*c.*80% of total N) were higher than

the cattle slurry (mean 54% of total N), with the solid materials containing <10% of the total N in the $\text{NH}_4\text{-N}$ form.

3.2 Ammonia emissions

Ammonia emissions from the spring 2012 applications at NW were similar from the food-based digestate and cattle slurry (23-31% total N applied), and both these treatments had higher emissions than the green compost and cattle FYM (<5% total N applied; $P<0.05$). Perhaps surprisingly, there were no differences ($P>0.05$) in NH_3 emissions between the bandspread and broadcast food-based digestate and cattle slurry treatments.

At PW, NH_3 emissions following the spring 2012 organic material applications were higher from food-based digestate (48-63% total N applied) than from cattle slurry (12-37% total N applied), with the green compost and cattle FYM treatments having much lower emissions (<3% total N applied; $P<0.05$). Bandspreading reduced NH_3 emissions from the food-based digestate by c.25% (although this was not significant; $P>0.05$) and cattle slurry by 70% ($P<0.05$) compared with the respective broadcast applications.

At WE, NH_3 emissions following the autumn 2011 organic material applications were again greater from food-based digestate (56-60% total N applied) than from pig slurry (c.40% total N applied), with the pig FYM and green/food compost having the lowest emissions (4-14% total N applied; $P<0.05$). There were no significant differences in NH_3 emissions between the bandspread and broadcast (soil incorporation within 24 hours) food-based digestate and pig slurry applications probably because the bandspread digestate and slurry stayed on the soil surface due to the dry (hydrophobic) soil conditions. NH_3 emissions were lower following the spring 2012 food-based digestate and pig slurry applications than in autumn because the soil was moist leading to more rapid soil infiltration compared with the dry, hydrophobic soil surface of the arable stubble in autumn which restricted infiltration. Ammonia emissions were similar from the food-based digestate and pig slurry treatments (c.20% total N applied), with the pig FYM and green/food compost again having lower emissions (c.2% total N applied; $P<0.05$). There were no significant differences between the bandspread and broadcast food-based digestate and pig slurry treatments, which was most probably due to the bandspread applications not staying in a band (plus some temporary ponding on the soil surface), followed by 'rapid' infiltration into the soil on both treatments.

The cross-site analysis of the total NH_3 losses from the autumn 2011 applications at WE, and the spring 2012 organic material applications at NW, PW and WE is summarised in Table 3. Ammonia emissions were similar on the broadcast and bandspread food-based digestate (c.40% of total N

applied) and were greater than those from the broadcast and bandspread livestock slurry, with FYM and compost having the lowest emissions. There was no difference in emissions due to spreading method for the food-based digestate, but broadcast slurry gave rise to greater emissions than bandspread slurry. Total NH_3 losses from food-based digestate were of a similar magnitude to the 35-65% of total N applied reported by Tiwary *et al.* (2015).

The cross-site NH_3 emission curves (Figure 1) indicate that the majority of the NH_3 losses from food-based digestate and livestock slurry occurred within 6 hours of spreading.

3.3 Nitrous oxide emissions

Nitrous oxide fluxes at NW in spring 2012 were generally low throughout the measurement period at $<20 \text{ g N}_2\text{O-N/ha/day}$ on all treatments (Figure 2a). There was a small peak in emissions ($c.30 \text{ g N}_2\text{O-N/ha/day}$) at the end of May which corresponded with a drop of $c.10\%$ in the WFPS. Cumulative N_2O losses (net of the control) were all low ($<0.5 \text{ kg N}_2\text{O/ha}$) and there were no significant differences in EFs between the different treatments (Figure 2b).

At PW in spring 2012, N_2O fluxes were generally higher than at NW. Emissions peaked (particularly on the FYM treatment at $c.110 \text{ g N}_2\text{O-N/ha/day}$) around 2 weeks after the organic materials were applied in early May; after this, emissions on all the organic material treatments returned to background levels ($c.10 \text{ g N}_2\text{O-N/ha/day}$; Figure 3a). As at NW, cumulative N_2O losses (net of the control) were all low ($<0.5 \text{ kg N}_2\text{O/ha}$). Emissions from the green compost treatment were significantly lower than from the cattle slurry and FYM treatments ($P<0.05$). Although bandspreading significantly increased the EF from cattle slurry compared with surface broadcasting ($P<0.05$), there was no difference in the EF between the broadcast and bandspread digestates. However, the bandspread digestate had a significantly ($P<0.05$) lower EF than the bandspread cattle slurry (Figure 3b).

Nitrous oxide fluxes at WE in autumn peaked at $c.100 \text{ g N}_2\text{O-N/ha/day}$, shortly after the organic materials were applied in early August 2011; emissions on all the organic material treatments had returned to background levels ($c.10 \text{ g N}_2\text{O-N/ha/day}$) by the end of November 2011 (Figure 4a). Net cumulative N_2O losses ranged from $0 \text{ kg N}_2\text{O/ha}$ on the compost treatment to 1.2 kg/ha on broadcast digestate treatment. Although the EF for green/food compost was very low, and bandspreading slurry and digestate resulted in numerical reductions in the EF, none of the treatment effects were significant (Figure 4b).

At WE in spring, N_2O fluxes peaked at $c.70 \text{ g N}_2\text{O-N/ha/day}$ in March about 1 month after the organic materials were applied in late February 2012; emissions on all the organic material treatments had

returned to background levels (c.10 g N₂O-N/ha/day) by the end of April 2012 (Figure 5a). Net cumulative N₂O losses ranged from <0 kg N₂O/ha on the compost treatment to 1.8 kg/ha on the bandspread digestate treatment. Emissions from the green compost were significantly lower than from the digestate treatments and the bandspread pig slurry ($P<0.05$). There was no effect of bandspreading on the EF for digestate or pig slurry compared with surface broadcasting (Figure 5b). Overall, there was no significant effect of food-based digestate/livestock slurry application method (i.e. surface broadcast compared with bandspreading) on N₂O emissions (Table 3).

3.3.1 Leaching losses

Drainage volumes at WE over the winter of 2011-12 were low (92 mm) due to the lower than average over-winter rainfall of c.200mm compared to the 25 year average of just over 300 mm. Nitrate concentrations in the drainage water were c.50 mg/l on all treatments at the start of drainage in November/December 2011; concentrations peaked in January/February 2012 and were highest (c.150 mg/l) on the surface broadcast food-based digestate treatment (Supplementary Information Figure S1a). The peak in leachate NO₃⁻ concentrations occurred somewhat later than the maximum soil NO₃⁻ concentrations which were measured in September 2011, shortly after the organic materials were applied (Supplementary Information Figure S1b). This was probably because the NO₃⁻ present in the top 15cm of the soil took this time to move through the soil profile to 30cm, which was the depth at which the leachate was sampled; additional NO₃⁻ would also be mineralised over the autumn period from the organic N applied with the organic materials, contributing to NO₃⁻ leaching losses.

Cumulative NO₃⁻ leaching losses following the food-based digestate and pig slurry treatments were greater ($P<0.05$) than from the pig FYM and compost treatments, with no significant differences between bandspread and broadcast food-based digestate treatments (Figure 6). Ammonium-N concentrations in the drainage waters were very low on all treatments (<0.05 mg/l) and cumulative leaching losses were <0.02 kg/ha (i.e. <0.01% of the total N applied).

The winter wheat crop (drilled in early October) was only expected to take up 5-10 kg N/ha during the autumn growth phase which effectively ceases by the end of October (Nicholson *et al.*, 2013). This is equivalent to <10% of the total N applied with the organic materials and hence would have had little effect on the quantities of N leached over the winter period from the different organic materials.

4. Discussion

4.1 Ammonia emissions

The higher NH_3 emissions from the food-based digestate than from livestock slurry ($P < 0.05$) were most probably due to the greater $\text{NH}_4\text{-N}$ content of the food-based digestate (mean 5.6 kg/t) compared with the livestock slurries (mean 2.2 kg/t for pig slurry and 1.3 kg/t for cattle slurry), Table 2. Additionally, the mean pH of the food-based digestate was 8.5 compared with 7.8 for pig slurry and 7.6 for cattle slurry (Table 2). It is known that pH values greater than 8 are particularly conducive to elevated NH_3 emissions from digestates (e.g. Hoeksma *et al.*, 2012) and digestion of livestock slurry has been shown to increase pH with a concomitant increase in NH_3 volatilisation (Sommer *et al.*, 2006). Acidification (i.e. decreasing the pH) has been adopted as the Best Available Technology (BAT) for reducing NH_3 losses from livestock slurry in some European countries (Kai *et al.*, 2008), however further research into the costs, practicalities and effectiveness of acidification of food-based digestates as a method of controlling NH_3 emissions is still required.

Overall, bandspreading was effective at reducing NH_3 emissions from livestock slurry compared with surface broadcasting as reported in many previous studies (e.g. Sommer *et al.*, 1997; Webb *et al.*, 2005), because the bandspreading technique reduces the surface area of slurry exposed to the atmosphere from which NH_3 can be volatilised. However in this study bandspreading was not found to be an effective mitigation option for food-based digestate. Bandspreading of liquid organic materials (such as food-based digestate) is now a common practice, with the majority of contractor-spread digestate applied using bandspreaders. In this study, the failure to observe a reduction in NH_3 emissions when bandspreading food-based digestate (in comparison with surface broadcast applications) was most probably due to soil and/or organic material properties that meant that the digestate did not rapidly infiltrate into the soil or did not stay in a narrow band on the soil surface. Dry matter content is known to affect NH_3 emissions from cattle slurry, with emissions increasing as slurry dry matter content increases (e.g. Sogaard *et al.*, 2002; Misselbrook *et al.*, 2004); it is likely that this relationship will also hold when comparing food-based digestates with different dry matter contents, although we do not know of any research data specific to digestates. The importance of crop height as a factor affecting NH_3 emissions is not clear; previous UK/Danish research has shown that the abatement efficiency of bandspread slurry applications increased with grass height and was typically 60% (Thorman *et al.*, 2008). In contrast, more recent Irish data (Lalor *et al.*, 2012) has not supported a relationship between NH_3 emission reductions from bandspread slurry applications and grass height. There is a requirement for further research on this topic to enable us to better assess the importance of crop height in controlling NH_3 losses from food-based digestates. It is important to

bear in mind that bandspreading technologies provide numerous other advantages over broadcast applications (e.g. more even digestate application and hence more accurate assessment of application rates, the ability to apply from tramlines, reduced odour and crop damage, and a cleaner sward) implying that it is still the best application method available.

In line with findings from previous studies using livestock slurry (Nicholson *et al.*, 2013), Figure 1 indicates that the majority of the NH_3 losses from food-based digestate occurred within 6 hours of spreading. In contrast, Tiwary *et al.* (2015) found that NH_3 losses from food based digestates continued for about 2 days following spreading (although NH_3 emission patterns in tropical climates are likely to be different from those in temperate conditions), but that incorporation immediately following application reduced NH_3 losses by 85%. These findings highlight the importance of rapid soil incorporation as an alternative method for preventing N losses via this pathway on arable (cultivated) soils. Rapid soil incorporation has previously been shown to reduce NH_3 losses following slurry applications (e.g. Sommer & Hutchins, 2001; Webb *et al.*, 2005), although UK survey results have shown that a significant proportion of cattle and pig slurry (c.65%) applied to tillage fields is either not incorporated, or only incorporated more than 24 hours after spreading (Benford, 2016). This study has demonstrated that rapid incorporation could have the same mitigation potential when used with food-based digestate. However, the technique has also been shown to increase N_2O emissions following slurry application as (Thorman *et al.*, 2011) suggesting that the potential for so called 'pollution swapping' could also exist when it is used with digestates.

In addition to representing the loss of a valuable resource, NH_3 emissions from digestate applications present a challenge to the UK meeting EU directives on NH_3 emissions. Under the EU National Emissions Ceiling Directive, the UK has a proposed target to reduce ammonia emissions by 8% (relative to a 2005 baseline) between 2020 and 2029, and by 21% from 2030. The UK Ammonia Emissions Inventory (UKAEI) includes emissions from food and crop-based digestates (but in the non-agricultural Inventory), although better data are needed to improve the current estimates. Based on the emission factors measured in this study (i.e. 38% for bandspread food-based digestate; Table 3) and the estimated 1.4 million m^3 of food-based digestate currently applied to agricultural land (WRAP, 2014) with an average total N content of 5 kg/m^3 , this implies that food-based digestate will emit an additional 3.3 kt of NH_3 , equivalent to 1% of the UK emission target (297 kt for 2010). If this is scaled up to the predicted 2020 production volumes (4.25 million m^3), then food-based digestate would emit an additional 9.8 kt of NH_3 or 3.3% of the UK 2010 target. However, these estimates do not account for reductions in ammonia emissions from manufactured (mineral) N fertiliser use.

4.2 Nitrous oxide emissions

Nitrous oxide emissions from agricultural soil are predominately produced via the microbially mediated processes of nitrification and denitrification (Firestone & Davidson, 1989). The factors which control the magnitude of N₂O emission include soil mineral nitrogen (SMN) content, soil temperature, soil moisture content (Dobbie & Smith, 2001; 2003) and available carbon (Weier *et al.*, 1993). Davidson (1991) showed that there is an optimum production of N₂O, which occurs at a water-filled pore space (WFPS) roughly equal to a transition point below which N₂O is predominantly emitted from the aerobic process of nitrification and above which N₂O is predominantly emitted from the anaerobic process of denitrification. Davidson (1991) suggested that this transition occurs at a WFPS of 60%; other studies, however, have shown that the position of the maximum emission can vary with soil type and conditions. Notably, UK studies have indicated that the highest N₂O emissions frequently occur as a result of the anaerobic process of denitrification i.e. at a WFPS >60% (e.g. Dobbie & Smith, 2001; 2003).

At all the 3 experimental sites, most N₂O emissions occurred in the few weeks following the organic materials being spread to land, and had generally returned to background levels within c.2 months (Figures 2a, 3a, 4a and 5a). Tiwary *et al.* (2015) also reported that N₂O emission rates were “feeble” past 24 hours following spreading of food-based digestate at a site in India, and emissions from soil incorporated digestate were negligible; however they only measured emissions for 30 days.

At NW and WE, the highest N₂O emissions corresponded to a peak in soil NH₄-N concentrations (see Supplementary Information, Figure S2) suggesting that N₂O was being produced as a result of the nitrification of the NH₄-N in the applied organic materials to NO₃⁻-N by soil micro-organisms. In this study, there was no clear relationship observed between N₂O emission rates and changes in the WFPS (Figures 2a, 3a, 4a and 5a). However, at NW there was a peak in emissions corresponding to a c.10% drop in WFPS which may have been caused by a change from anaerobic to aerobic conditions which moved the end product of denitrification from N₂ to N₂O.

Importantly, N₂O emission factors (EFs) from all the organic material treatments were below the IPCC Tier 1 default value of 1% of total N applied, and in the case of compost was not significantly different from background values, although the variability associated with emissions from some treatments suggests that the IPCC value Tier 1 default value could be exceeded on some occasions (see Figures 3b and 5b). The EF for food-based digestate (mean 0.45% ± 15%) was much lower than those obtained by Tiwary *et al.* (2015) of 4 – 10%, although the latter values were obtained under tropical climatic conditions which are very different to the UK and may have promoted N₂O losses. Data derived from the present study could be used in conjunction with other recently published research results on N₂O emissions from fertilisers and manures (Bell *et al.*, 2015a,b; Bell *et al.*, 2016)

to reduce some of the uncertainty in the UK national agricultural N₂O inventory through the generation of robust and experimentally verified Tier 2 EFs in compliance with the requirements of the IPCC (IPCC, 1996; 2006).

4.3 Nitrate leaching

Nitrate leaching losses from autumn food-based digestate applications were c.15% of the total N applied compared with the IPCC default value of 30% for all agricultural soil N sources (i.e. synthetic fertilisers, manures, compost, sewage sludge, crop residues and other organic N sources), which is used when calculating indirect N₂O losses from N applications to land. Research in the 1990s using livestock manures showed that NO₃⁻ leaching losses can be greatly reduced by applying the materials in spring compared with autumn applications (e.g. Chambers *et al.*, 2000), and this has led to the introduction of no-spreading periods for high available N materials being an integral part of NVZ legislation in England, Scotland and Wales (SI, 2015; SSI, 2013; WSI, 2013). Despite this, recent UK survey data have shown that significant percentages of biosolids (64%) and other non-farm organic materials (32%) are still applied to winter sown crops in August, September and October (Benford, 2016) and hence may be prone to nitrate leaching losses. This is mainly for practical reasons due to storage capacity pressures, and so that farmers can spread materials to land while soils are still trafficable and before a crop is sown. The results from our study strongly suggest that, as for livestock slurry and other high readily available N organic materials, farmers should be advised to apply food-based digestate in the spring where practically possible, or in autumn to an actively growing crop such as grass or oilseed rape which will take up available N from the soil so it will not be lost via overwinter NO₃⁻ leaching.

5. Conclusion

The results of this study have shown that N losses via NH₃ volatilisation following land spreading of food-based digestate were high at both the arable and grassland sites (30-50% of total N applied). This was due to the high readily available N content and high pH of the food-based digestate (mean 8.5), and the soil conditions at the time of spreading which affected the rate at which the materials infiltrated into the soil matrix. Precision application (i.e. bandspreading) can reduce NH₃ emissions, but the effectiveness of these techniques is dependent on the prevailing soil conditions. Because the majority of the NH₃ losses occurred within 6 hours of spreading, it is important that where possible farmers are encouraged to rapidly incorporate food-based digestate into the soil as a method for conserving N so it can be utilised by the crop (assuming application to spring crops). In contrast, N₂O

losses from food-based digestate were low, with measured emission factors all less than the 1% IPCC default value (mean $0.45\% \pm 0.15\%$). There was no significant difference between N_2O losses from food-based digestate applied by surface broadcast or bandspread techniques; however, we would recommend that further research is undertaken on different soil types and under different climatic conditions to confirm this finding, as these are important factors controlling denitrification/nitrification processes. Autumn applications could potentially lead to a total of 75% of the applied N being lost by ammonia volatilisation (c.60%) and leaching (c.15%), constituting the loss of a valuable resource with consequent economic and environmental implications. This strongly suggests that farmers should be advised to apply these materials in the spring where practically possible. Emissions (NH_3 , N_2O , NO_3^-) from green compost were all low, indicating that in these terms compost can be considered as a low risk material in terms of N losses, which can be used to build up soil long-term (organic) N reserves and to improve soil condition.

The information produced from this study has been used to develop best practice guidelines for digestate and compost use in the UK that seek to maximise crop nutrient utilisation and to minimise emissions to the environment (WRAP, 2016). Furthermore, the N_2O and NH_3 EFs derived in this study for both food-based digestate and composts could be used to improve the estimates of emissions from these sources in the UK GHG and Ammonia Emission Inventories, although emissions under other agroclimatic conditions are likely to be different.

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724 **List of Tables**

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726 Table 2. Organic material analysis at each site

727 Table 3. Cross-site NH₃ and N₂O emission factors. *Means labelled with different superscript letters*
728 *are significantly (P<0.05) different from each other.*

729

730 **Table 1. Baseline soil characteristics, rainfall and cropping at the experimental sites.**

Site code	Site name and location	Soil texture	Clay (%)	Total N (%)	Organic C (%)	pH	Annual rainfall (mm)	Cropping
NW	North Wyke (Devon, England)	Clay loam	38	0.4	6.9	5.5	1,031	Grassland
PW	Pwllpeiran (Ceredigionshire, Wales)	Clay loam	28	0.5	4.7	5.1	975	Grassland
WE	Wensum (Norfolk, England)	Sandy loam	11	0.2	1.3	6.7	594	Winter wheat

731

732

733 **Table 2. Organic material analysis, application rates and N loadings at each site**

Determinand	Food-based digestate	Compost*	Slurry**	FYM**
Site: NW				
Dry matter (%)	5.1	60	6.1	20
Total N (kg/t fw)	8.0	14	2.6	5.8
NH ₄ -N (kg/t fw)	5.8	0.7	1.4	<0.1
NO ₃ ⁻ -N (kg/t fw)	<0.1	<0.1	<0.1	<0.1
NH ₄ -N (% total N)	73	5	56	<1
pH	8.1	8.5	8.2	9.0
Application rate (t/ha)	20	20	30	25
N loading (kg/ha)	160	271	77	144
Site: PW				
Dry matter (%)	6.1	51	4.9	24
Total N (kg/t fw)	5.4	7.0	2.2	4.9
NH ₄ -N (kg/t fw)	3.9	<0.1	1.2	<0.1
NO ₃ ⁻ -N (kg/t fw)	<0.1	<0.1	<0.1	0.4
NH ₄ -N (% total N)	72	1	53	8
pH	8.4	7.9	7.0	7.3
Application rate (t/ha)	20	20	30	25
N loading (kg/ha)	107	140	67	122
Site: WE autumn				
Dry matter (%)	5.4	54	2.3	24
Total N (kg/t fw)	7.8	11	3.0	8.1
NH ₄ -N (kg/t fw)	6.3	1.5	2.2	0.8
NO ₃ ⁻ -N (kg/t fw)	<0.1	<0.1	<0.1	0.1
NH ₄ -N (% total N)	82	14	75	9
pH	8.8	7.0	7.5	8.4
Application rate (t/ha)	32	20	41	30
N loading (kg/ha)	245	216	122	244
Site: WE spring				
Dry matter (%)	4.4	48	2.7	23
Total N (kg/t fw)	6.9	9.0	2.6	9.2
NH ₄ -N (kg/t fw)	6.2	0.2	2.2	0.2
NO ₃ ⁻ -N (kg/t fw)	<0.1	<0.1	<0.1	0.6
NH ₄ -N (% total N)	89	2	86	3
pH	8.7	8.3	8.0	7.4
Application rate (t/ha)	30	20	38	30
N loading (kg/ha)	207	181	98	277

734 fw = fresh weight

735 *Green compost at NW and PW; green/food compost at WE

736 **Cattle slurry and FYM at NW and PW; pig slurry and FYM at WE

737 **Table 3. Cross-site NH₃ and N₂O emission factors. Means labelled with different superscript letters**
738 **are significantly ($P<0.05$) different from each other.**

Treatment	Mean NH ₃ EF	Mean N ₂ O EF
	(% total N applied)	
Food based digestate – surface broadcast	42 ^d	0.47 ^b
Food based digestate – bandspread	38 ^d	0.43 ^b
Livestock slurry – surface broadcast	31 ^c	0.35 ^b
Livestock slurry – bandspread	24 ^b	0.55 ^b
Livestock FYM	4.5 ^a	0.28 ^b
Compost	3.3 ^a	<0.01 ^a
<i>SED</i>	3.3	0.12
<i>LSD</i>	6.7	0.24

739 SED: Standard error of difference of means

740 LSD: Least significant difference of means (5% level)

741

List of figures

Figure 1. Cross-site ammonia emissions curves.

Figure 2. NW spring: a) daily mean N₂O fluxes and b) N₂O emission factors. *Error bars show the standard error; SED = standard error of difference of means. No significant treatment differences*

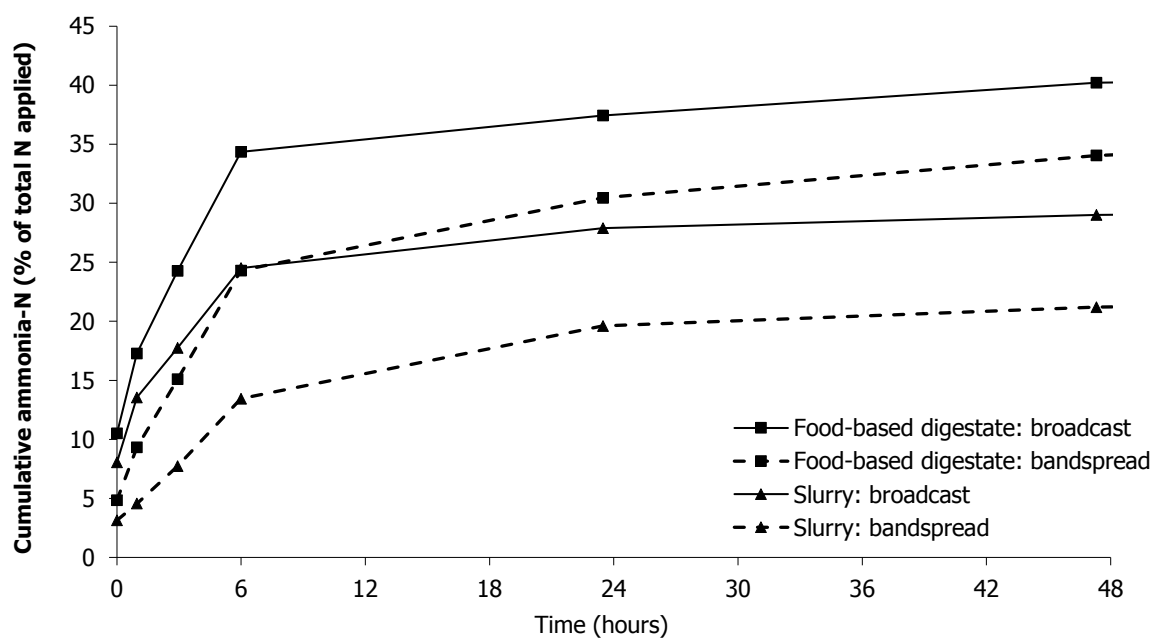
Figure 3. PW spring: a) daily mean nitrous oxide fluxes and b) emission factors. *Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly ($P<0.05$) different from each other.*

Figure 4. WE autumn: a) daily mean N₂O fluxes and b) N₂O emission factors. *Error bars show the standard error; SED = standard error of difference of means. No significant treatment differences.*

Figure 5. WE spring: a) daily mean N₂O fluxes and b) N₂O emission factors. *Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly ($P<0.05$) different from each other.*

Figure 6. Leaching losses (% of total N applied) following the autumn 2011 organic material applications. *Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly ($P<0.05$) different from each other.*

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760 **Figure 1. Cross-site ammonia emissions curves.**

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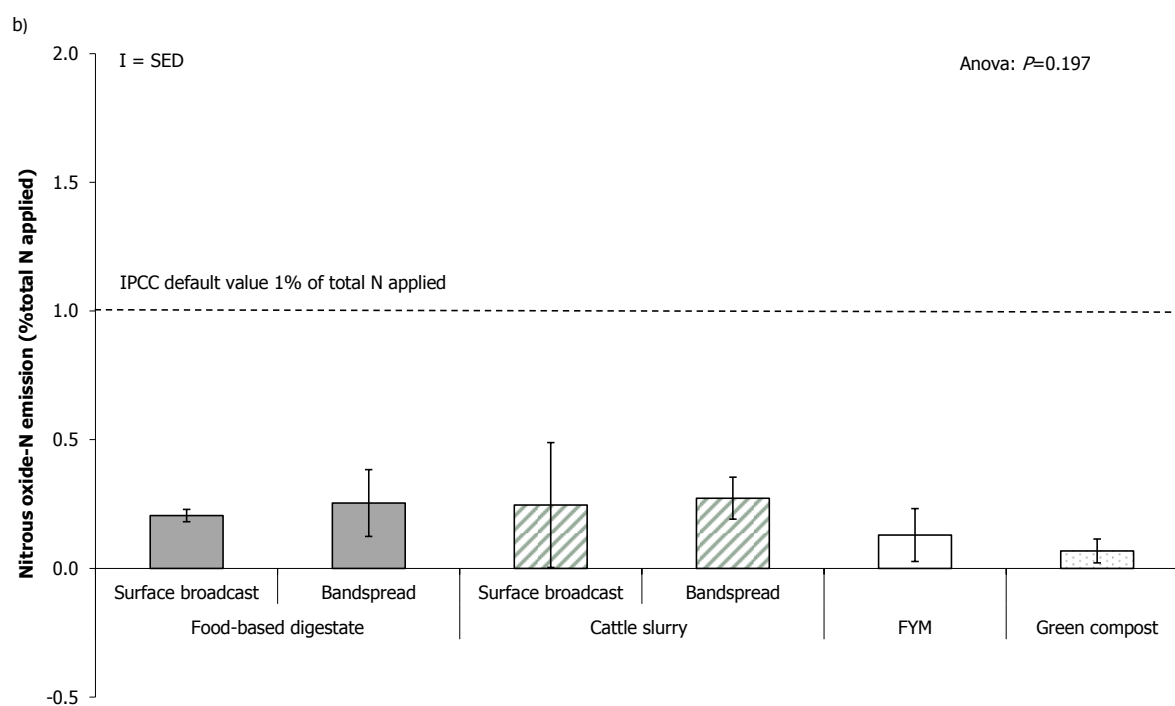
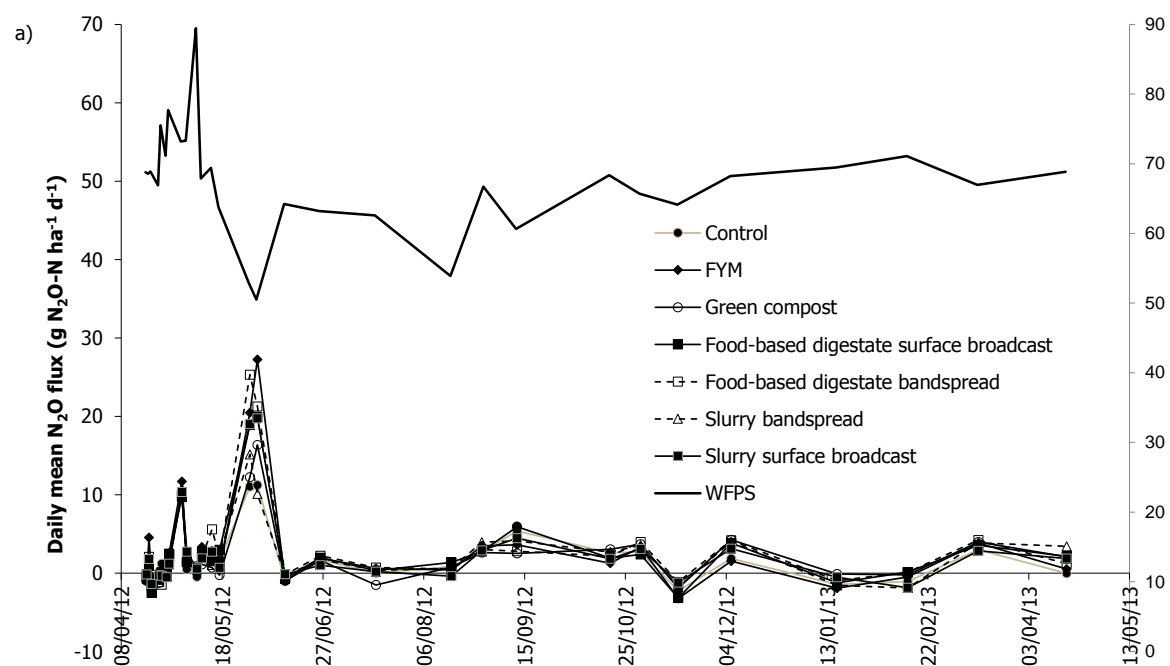


Figure 2. NW spring: a) daily mean N_2O fluxes and b) N_2O emission factors. Error bars show the standard error; SED = standard error of difference of means. No significant treatment differences.

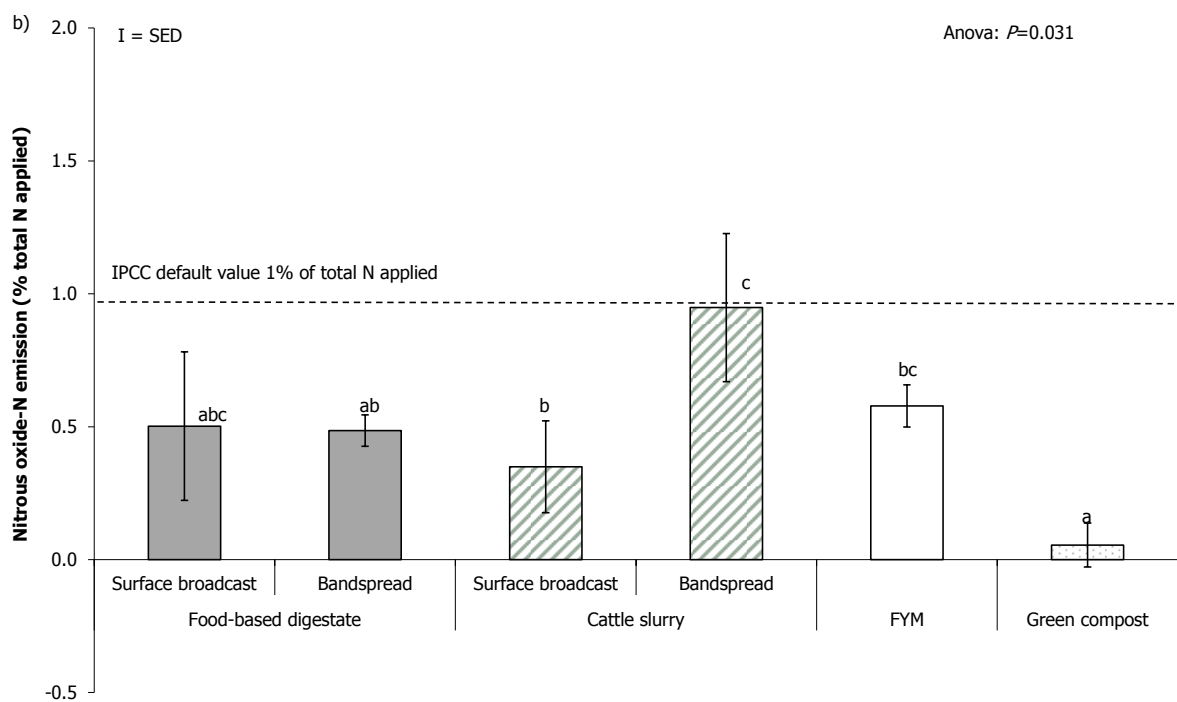
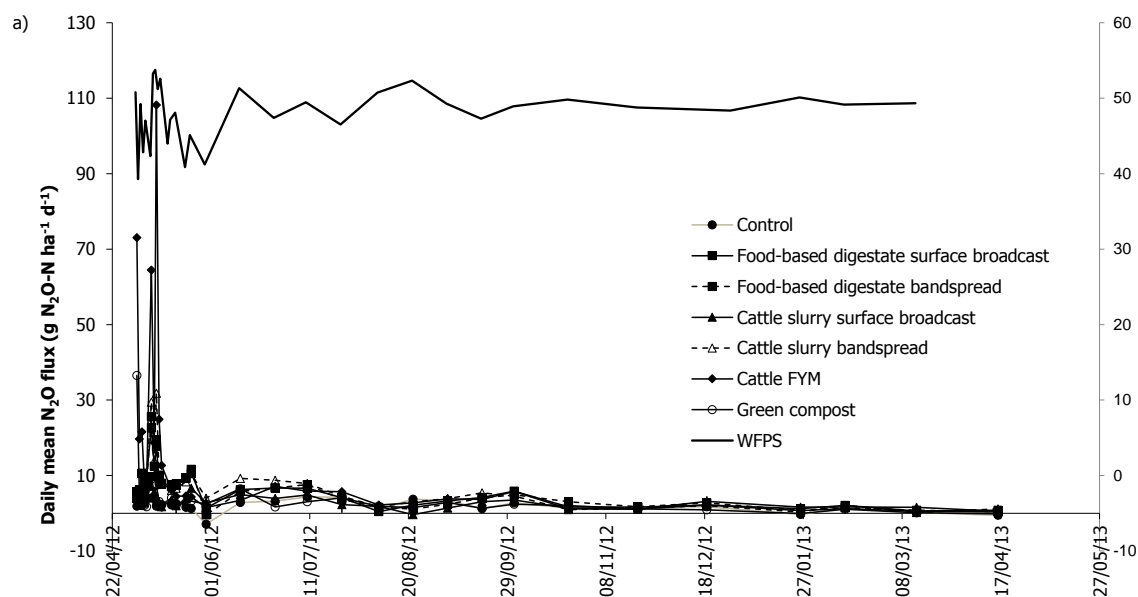


Figure 3. PW spring: a) daily mean nitrous oxide fluxes and b) emission factors. Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly ($P < 0.05$) different from each other.

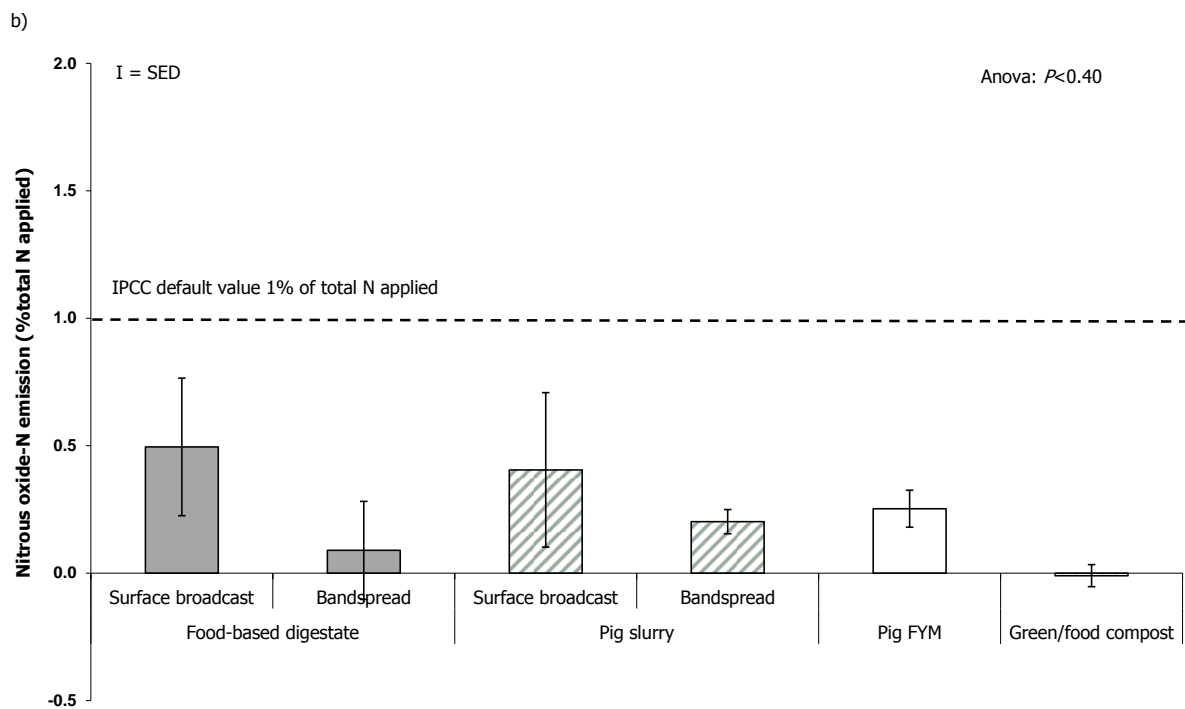
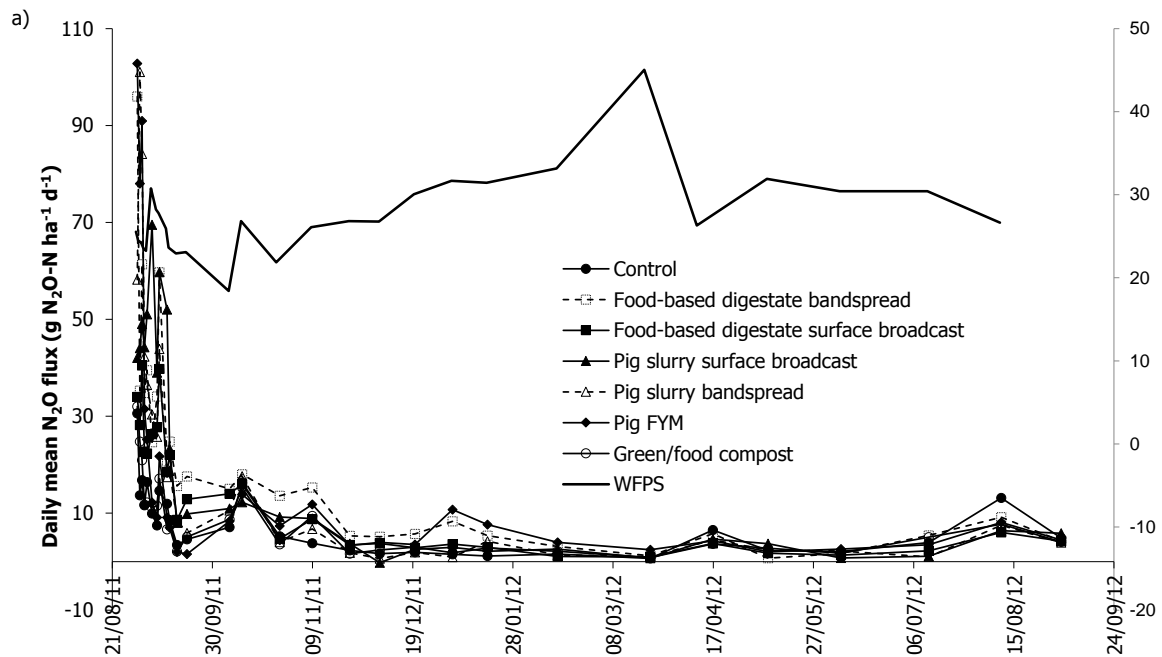
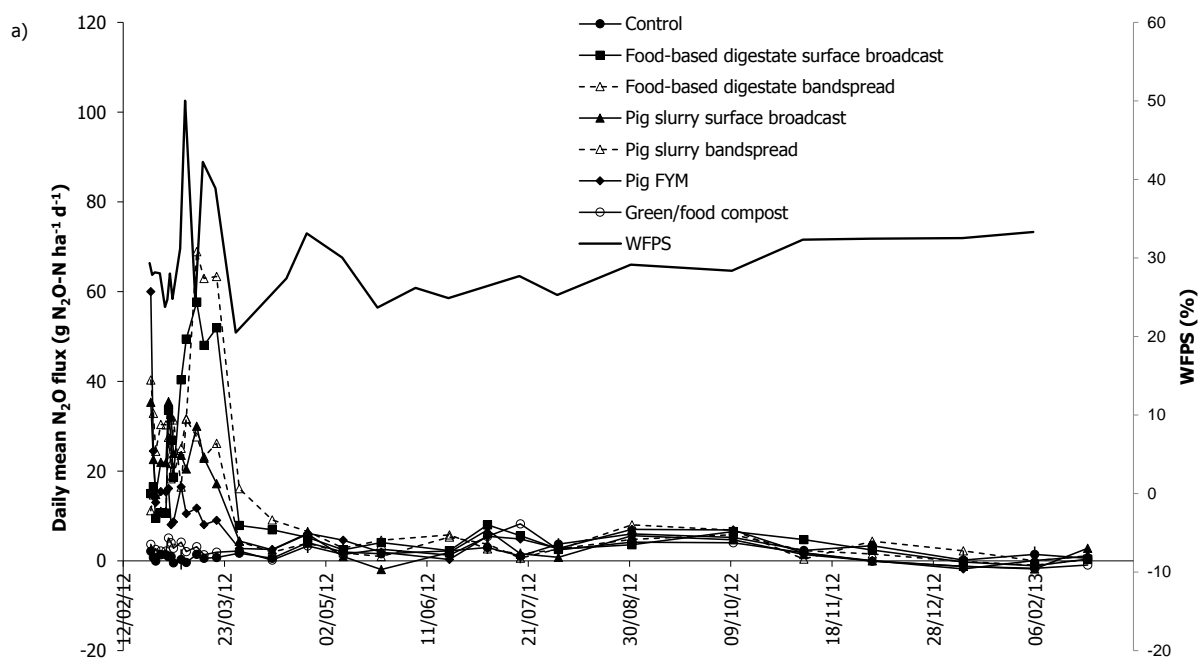


Figure 4. WE autumn: a) daily mean N_2O fluxes and b) N_2O emission factors. Error bars show the standard error; SED = standard error of difference of means. No significant treatment differences.



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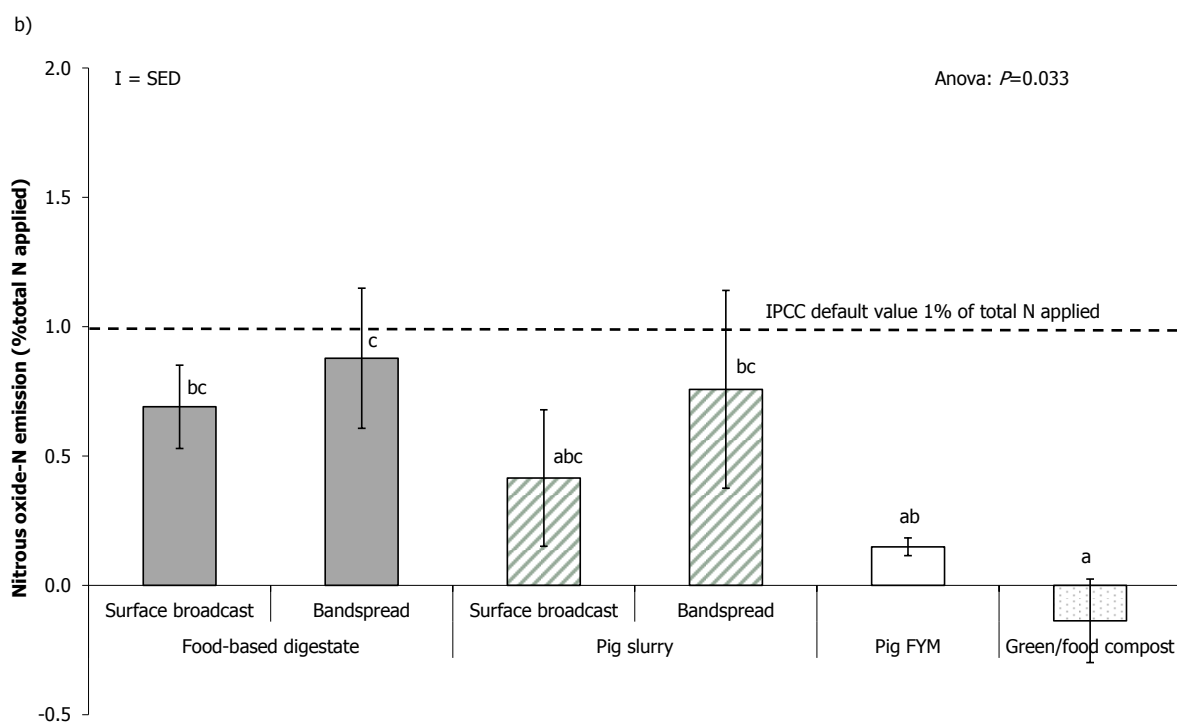


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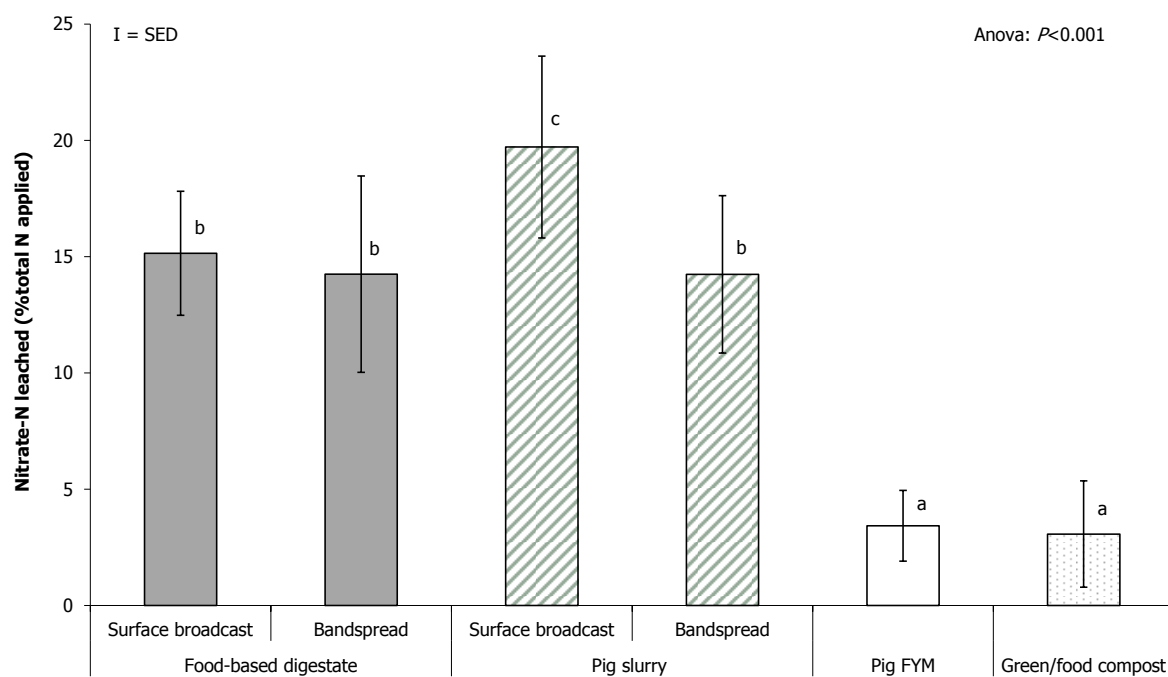


Figure 6. Leaching losses (% of total N applied) following the autumn 2011 organic material applications. Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly ($P < 0.05$) different from each other.

775 **Supplementary Information**

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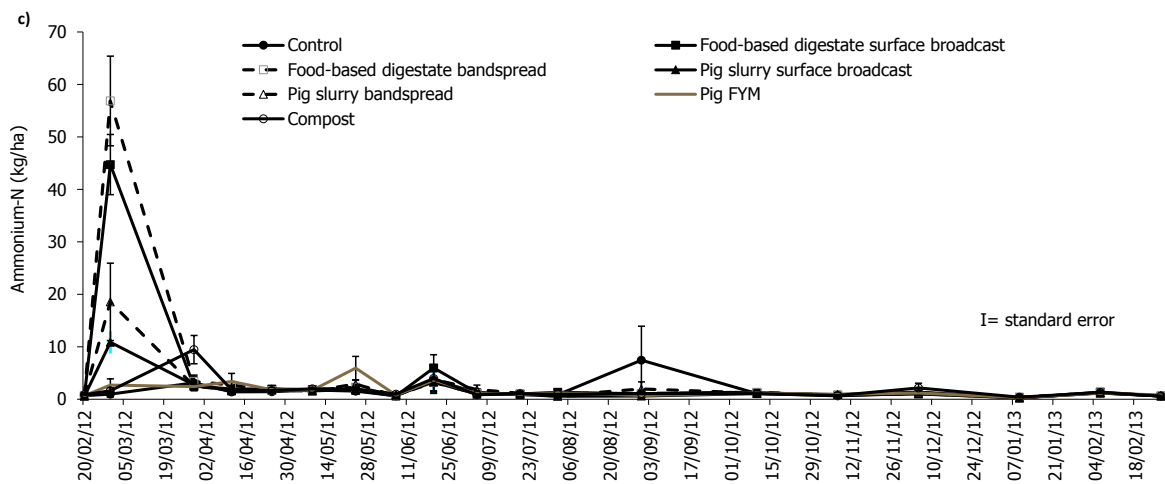
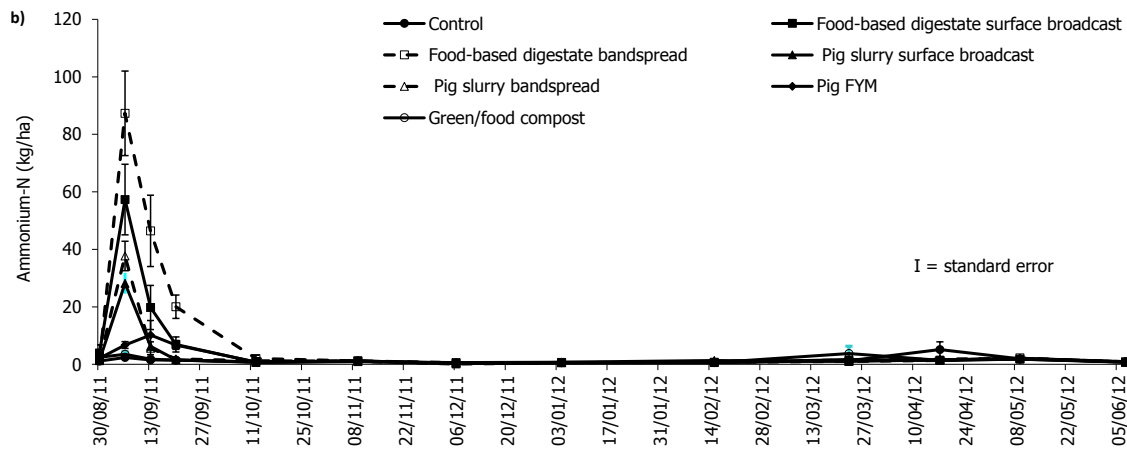
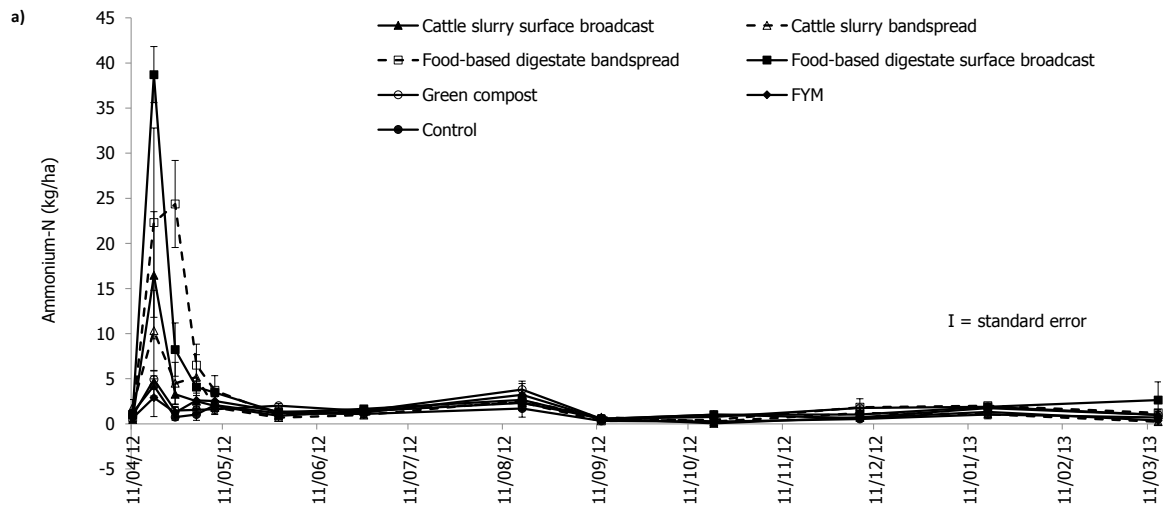


Figure S1. Soil $\text{NH}_4\text{-N}$ concentrations at a) NW, b) WE autumn and c) WE spring

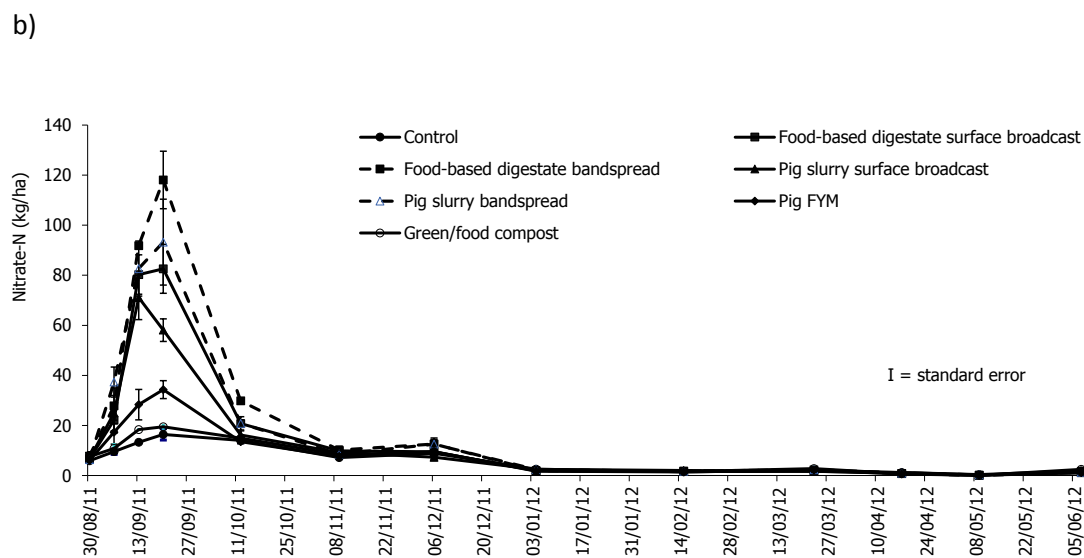
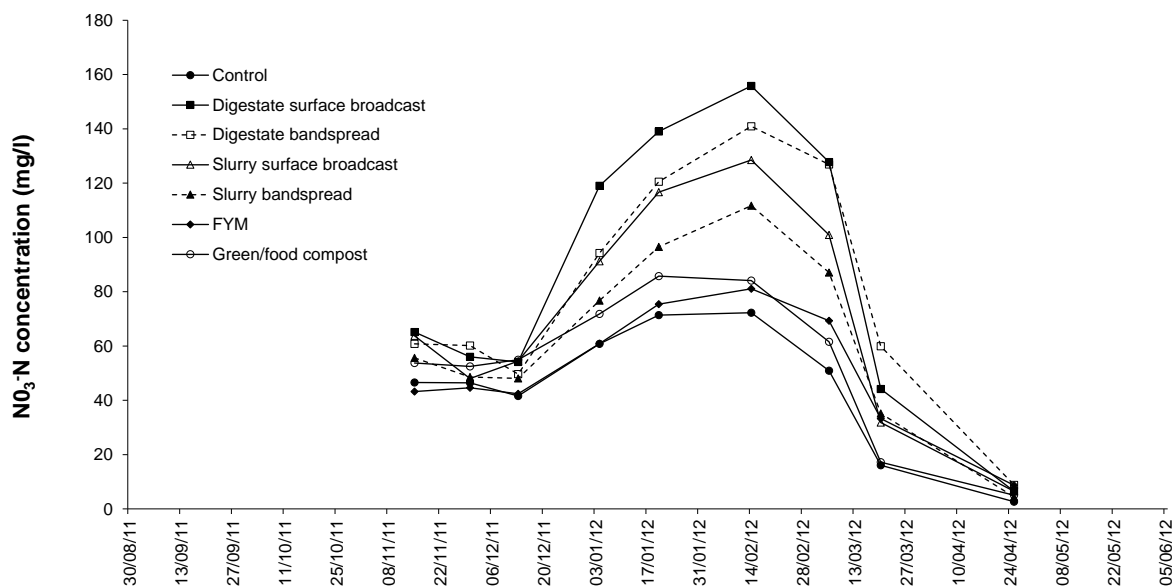


Figure S2. WE autumn a) NO_3^- concentrations in leachate following the autumn 2011 organic material applications and b) soil NO_3^- concentrations